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Post-fire carbon dioxide fluxes in the western Canadian boreal forest: evidence from towers, aircraft and remote sensing

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Abstract

Recent CO_2 flux measurements from towers and aircraft (net ecosystem exchange by eddy covariance) and remote sensing/modeling (net primary productivity—NPP) following fire show that the regenerating boreal forest in western Canada has a low initial flux that increases with time since fire. Daytime CO_2 fluxes are downward, even after 2 years following fire, although fluxes were upward during the first year after the fire. In summer, the forest is a net carbon sink a few years following fire. A regression of all data gives a relationship where the CO_2 flux relative to 10 years following fire $= 0.11 + 0.92 \log_{10}$ (years since fire) ($r^2 = 0.5$). The CO_2 flux reaches the same rate as that of a mature site between 10 and 30 years following fire, depending on the site and the data set. Many studies in the literature indicate that soil respiration decreases following fire, although several models assume that heterotrophic respiration increases. If fire reduces respiration and our growing season measurements showing a net sink in early years are widely applicable, it is likely that some models may have overestimated the impact of fire on the carbon balance of the boreal landscape.
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1. Introduction

Forests are an important part of the global carbon cycle (Dixon et al., 1994) and the boreal forest in particular is thought to have a strong influence on the net CO_2 exchange with the atmosphere (D'Arrigo et al., 1987; Bonan, 1991). Carbon budget models of the Canadian forest sector show trends of a net forest carbon sink through the first two-thirds of the 20th century, followed by a decreasing sink and even a carbon source in the latter part of the century (Kurz and Apps, 1999; Chen et al., 2000). This time trend in the model estimates of carbon flux is caused by a changing disturbance regime, mostly driven by increased fire and insect activity. A substantial amount of recent research has been devoted to understanding the impacts of fire on the forest carbon balance (e.g., Harden et al., 2000; Kasischke and Stocks, 2000). In Canada, about 2 million hectares of forest have burned annually on average from 1959 to 1999, with extreme fire years burning more than 7 million hectares

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(Weber and Stocks, 1998; Stocks et al., 2002). Mean direct carbon emissions from combustion in forest fires have been estimated at about 27 T_g C per year for this period (Amiro et al., 2001).

Canopy crown fires are common in the Canadian boreal forest and most of the area burned results in tree death. This regenerates the forest, creating a successional state that has different carbon dynamics than a mature forest. The belief is that in the early stages, photosynthesis is low where living vegetation is sparse and heterotrophic respiration is large because of available fire-killed vegetation as a substrate and warmer soil temperatures. This combination would tend to decrease the net carbon sink of the forest. However, we have few measurements of these processes. During the BOREAS experiment in western Canada in the 1990s, almost all measurements of CO_2 fluxes were made over relatively mature forests (Goulden et al., 1997; Jarvis et al., 1997; McCaughey et al., 1997; Sellers et al., 1997; Chen et al., 1999b), although some soil respiration measurements considered chronosequences (Burke et al., 1997; Zepp et al., 1997). Similarly, most current studies in other areas as part of the FLUXNET program focus on mature forests (Valentini et al., 2000). Short-term chronosequence flux studies in Siberia show the largest fluxes in a 67-year-old pine (*Pinus sylvestris*) stand compared to older or younger forests (Schulze et al., 1999). Recognizing the need for more information, there are several recent initiatives focusing on disturbed sites in North America.

Our recent studies have focused on estimating the effect of fire and, to some extent, harvesting on the forest CO_2 flux following the disturbance. We have tackled this at three different spatial scales. First, we have measured net ecosystem exchange (NEE) of CO_2 from towers above the forest using the eddy

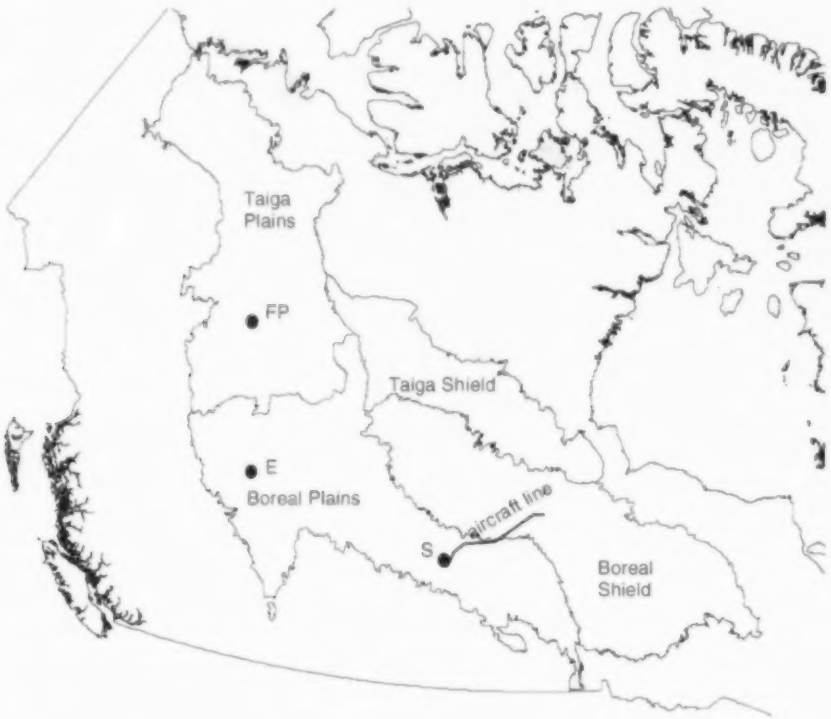

Fig. 1. Location of study sites. The sections of the four ecozones used for the NPP study are outlined. The aircraft transect is shown (the ends of the line are at 53.98 N, 104.79 W (near SSA) and 55.89 N, 98.00 W (near NSA)). The tower sites correspond to FP—Fort Providence, E—EMEND site, S—BERMS (Saskatchewan), with geographic coordinates given in Table 1.

covariance technique (Amiro, 2001). Second, we have analyzed data from the BOREAS study where NEE was measured by eddy covariance from aircraft over the landscape (Amiro et al., 1999). And third, we have used a combination of remote sensing and an ecosystem process model to estimate net primary productivity (NPP) for the boreal forest (Amiro et al., 2000). All these studies analyze trends with time since disturbance. However, a complete carbon balance requires an annual estimate of net biome exchange (NBE), which would incorporate fluxes over the whole forest. Ongoing work seeks to achieve this, but currently data are only available to look at partial components. In addition, we only have limited spatial and temporal data, which vary among the methods.

The goal of this paper is to amalgamate and compare data from the three different spatial scales to arrive at our best to estimate the impact of disturbance on the carbon flux. Such a relationship can then be used to help verify assumptions in carbon balance models and to suggest areas for future research. We have done this by restricting our analysis to an overlapping area that covers much of the boreal and taiga regions of western Canada (Fig. 1). We have also used the literature to suggest that some of the model assumptions related to the magnitude of post-fire carbon fluxes may need to be reconsidered, including the response of respiration.

2. Methods

2.1. Tower fluxes

2.1.1. Sites

Fluxes were measured at seven individual tower sites (Fig. 1). Six of these were paired treatments where recently disturbed and mature sites were compared simultaneously, whereas one disturbed site did not have a control site.

2.1.1.1. Paired measurements at burn-1-year site. The 1-year-old burned site was located to take advantage of studies at the International Crown Fire Modelling Experiment site near Fort Providence, NWT, where intense fires in jack pine (*Pinus banksiana*) stands were conducted (Alexander et al., 1998; Table 1). We measured on a 150 m × 150 m experimental plot surrounded by a cleared fire-guard area about 50 m wide exposing bare mineral soil (footprint issues are discussed by Amiro, 2001). The stand was severely burned on 9 July 1997 in about 5 min with an average fire spread rate of 0.6 m s^{-1} (Clark et al., 1999). A control site (labeled mature pine in Table 1) was selected about 700 m to the west of the burned plot in the undisturbed part of the forest. More than 1 km of fetch was available to the west, about 600 m fetch to the north and south, and 300 m to the east. The ground surface was mostly bearberry (*Arctostaphylos uva-ursi*), Labrador tea (*Ledum groenlandicum*) and lichens.

2.1.1.2. Burn-2-year site. The 2-year-old burn site was in the east part of Prince Albert National Park, Saskatchewan in the Waskesiu fire, which burned about 1700 ha in July 1998. The pre-fire forest consisted of jack pine and black spruce (*Picea mariana*) stands, with some intermixed aspen. The fire was severe consuming much of the top layer of organic soil and killing all trees. In 2000, much of the regenerating vegetation consisted of aspen (*Populus tremuloides*) saplings less than 1 m tall, with sparse grass and forbs. The site had limited fetch to the northeast sector and all data between 0 and 90° from north were excluded from the analysis. The site was in the boreal ecosystem research and monitoring sites (BERMS) region where mature control sites were operated by other research groups.

2.1.1.3. Paired measurements at burn-10-year site. The 10-year-old burn site was northeast of Prince Albert National Park, Saskatchewan (Table 1). This human-caused fire burned in 1989, covering about 13,500 ha. Parts of the area had been logged prior to the fire, and slash residues would have been burned in some locations. Parts of the area were aerially seeded with jack pine seeds in the winter of 1990. The present tree canopy was composed of balsam poplar (*Populus balsamifera*), jack pine, trembling aspen, and birch (*Betula papyrifera*) and prior to the fire, the stand consisted of these same species as well as black spruce. Numerous dead snags of black spruce and jack pine were still standing, although most had fallen over and formed a leaning mix of dry, dead tree boles. The understory vegetation consisted mostly of black spruce saplings, saplings of the tree overstory species, bearberry, blueberry (*Vaccinium myrtilloides*), raspberry

Table 1
Tower site characteristics

	Paired sites, Fort Providence, NWT		BERMS, Saskatchewan	Paired sites, BERMS, Saskatchewan		Paired sites, EMEND, Alberta	
	Burn-1-year	Mature pine	Burn-2-year	Burn-10-year	Mature pine-mixed	Harvest-1-year	Mature aspen
Stand age (years)	1	80	2	10	50	1	80
Latitude/longitude	61°35′N, 117°10′W	61°35′N, 117°10′W	54°05′N, 106°00′W	54°15′N, 105°53′W	54°12′N, 105°59′W	56°45′N, 118°20′W	56°45′N, 118°20′W
Period	7–15 July 1998		12–27 July 2000	10–26 August 1999		13–22 July 1999	
Fetch (km)	0.1	>1	>1	>1	>1	0.1–0.3	>0.3
Stand type	Jackpine, black spruce	Jackpine, black spruce	Jackpine, spruce, aspen	Jackpine, spruce, aspen, poplar	Jackpine, spruce, aspen, poplar	Aspen, poplar	Aspen, poplar
Tree density (stems/ha)	6000 dead	3600 live, 2700 dead	1100	4300	2700	50,000	700
Canopy height (m)	13 (dead, leafless)	13	15–20 (dead, leafless)	3.5	15	1.5	20
Tree dbh (cm)	5.4	9.5	15	4	11	–	24
Instrument height (m)	3	18	8	6	18	3	24
Mean air temperature (°C)	20		17	19		16	
Mean daily C flux ($gC\,m^{-2}$ per day)	+0.8	−0.5	+0.01	−1.3	−1.3	+1.6	−3.8
Energy balance correction	1.11	1.07	1.0	1.23	1.32	1.38	1.07
No. half-hour periods for CO_2 flux	268	268	717	359	359	374	374

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(*Rubus idaeus*), rose (*Rosa acicularis*), bunchberry (*Cornus canadensis*), and reed grass (*Calamagrostis canadensis*). The control site (labeled mature pine-mixed) was selected to have about the same tree species mix as the burned site and was located about 9 km southwest of the burned site. The tree canopy was composed of jack pine, trembling aspen, black spruce, balsam poplar and white spruce (*Picea glauca*) with a sparse shrub layer of alder (*Alnus crispa*), gooseberry (*Ribes* spp.) and rose. Ground vegetation consisted mostly of bearberry, blueberry, bunchberry, Labrador tea, coltsfoot (*Petasites palmatus*), and false lily-of-the-valley (*Maianthemum canadense*). This area was mostly a mosaic of patches of about 100 m scale consisting of aspen, jack pine or black spruce.

2.1.1.4. Paired measurements at harvest-1-year site. The harvested treatment was located in conjunction with a large experimental program called Ecosystem Management by Emulating Natural Disturbance (EMEND; Spence et al., 1999). This experimental area is situated about 100 km northwest of Peace river, Alberta (Fig. 1). The general area is a mixed forest dominated by trembling aspen, balsam poplar, and white spruce. The treatment was a deciduous stand that was harvested during the winter of 1997/1998 as a full-tree harvest with the slash piled at the landing and burned during the following winter. By the summer of 1999, aspen sucker growth was about 1–1.5 m tall at a density of 5 stems m^{-2}. Other vegetation included fireweed (*Epilobium angustifolium*), rose, false lily-of-the-valley, gooseberry and reed grass. The surface had been mounded at about 3 m intervals resulting in mineral soil exposure in mounds about 0.3 m high, which were prepared for conifer seedling planting. However, the seedlings had not yet been planted when the flux measurements were made. The minimum fetch to a single line of standing trees was 100 m to the northwest and 300 m to the north, with full-canopied stands at distances of 300 m to the east and 170 m to the southwest. The control stand (labeled mature aspen in Table 1) was located about 1.3 km west of the harvested stand. This aspen/poplar plot covered an area of about 20 ha, with a minimum fetch of close to 300 m in most directions, but only about 120 m towards the east, from the tower location. The canopy understory consisted of rose, gooseberry, and buffaloberry (*Shepherdia canadensis*) shrubs, typically about 1 m tall. The ground vegetation layer was mostly vetch (*Vicia americana*), wintergreen (*Pyrola* spp.), bunchberry and fireweed.

2.1.2. Flux measurements

Triangular, 30 cm wide, towers were erected at each site and the flux measuring instruments were mounted on booms at the top, above the live canopy (Table 1). Three-dimensional wind velocities were measured with a CSAT3 sonic anemometer (Campbell Sci., Logan, UT), and the air-temperature fluctuations were measured using a 12.5 μm thermocouple within the axes of the anemometer to give the sensible heat flux. A 8 mm ID Bevaline IV tube was located within the axes of the sonic anemometer and the air stream was pulled down 3 m of this tube by a pump (Brailsford, Model TD-4X2, Rye, NY) at about 5 l min^{-1} and pushed through a 1 μm particle filter and the CO_2/H_2O analyzer (LICOR, Model 6262, Lincoln, NE). The CO_2 and H_2O concentrations were temperature and pressure corrected following McDermitt et al. (1994) as described by Chen et al. (1999b). Nitrogen flowed through the reference side of the gas analyzer at a rate of about 75 ml min^{-1} and the analyzer was operated in absolute mode. The gas analyzer was calibrated every few days by doing a zero check using nitrogen, and a span check using a known concentration (~360 ppm) of CO_2 from a cylinder, traceable to a reference standard. Water vapor span was based on the factory calibration and was checked in the field against a relative humidity sensor (Vaisala model HMI41, Helsinki, Finland) at ambient conditions. All instruments were powered by batteries and, in most cases, charged by solar panels. The signals from the anemometer (digital) and gas analyzer (fast analog) were sampled at 4 samples s^{-1} by a datalogger (Campbell Sci., Model 21X, Logan, UT) housed at the bottom of the tower. A constant time-lag of typically 0.75 s was determined through experimental tests, which did not change during the relatively short run of each experiment. Sufficient statistics were calculated every 30 min within the datalogger to allow post-processing of the data to determine corrected, rotated, fluxes using a covariance calculation (Tanner and Thurtell, 1969), although approximate fluxes were available on-line. We corrected the carbon flux data using average energy balance closure as described by Amiro (2001), with the correction factors shown in Table 1.

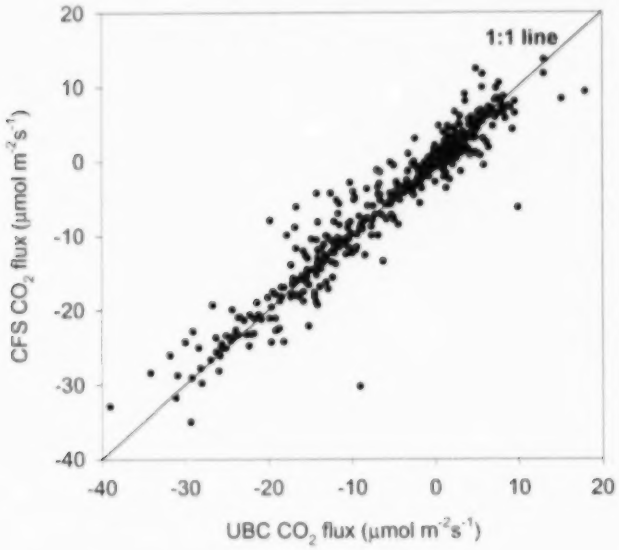


Fig. 2. Intercomparison between Canadian Forest Service (CFS) flux system used in the present paper and the University of British Columbia (UBC) flux system at the BERMS Old Aspen site, 10–22 July 2000. 1:1 line is shown, linear regression is $y = 0.94x - 0.19$, $r^2 = 0.92$.

Intercomparisons between our two flux systems were excellent with new tubing and filters, but degraded as the sample lines became dirty (Amiro, 2001). We also found good side-by-side agreement between one of our systems and the flux system run by the Black group of UBC at the present BERMS Old Aspen site (Fig. 2), except for some deviation near the end of the comparison, which we believe was caused by dirt in our lines. Over the 12-day-period, the mean carbon flux of the CFS system was 2.3% less than the UBC system. We believe that dirt was not an issue during the limited period of the flux measurements reported in this paper.

2.2. Aircraft fluxes

2.2.1. Flight lines

Data were collected during the BOREAS experiment intensive campaigns of June–August 1994 and 1996 (MacPherson, 1996; MacPherson and Bastion, 1997). These growing season data were selected to correspond to the same season as the tower flux data. We analyzed data collected during flights along a 500 km transect extending from near Prince Albert, Saskatchewan (53.98°N, 104.79°W) to near Thompson, Manitoba (55.89°N, 98.00°W) (Fig. 1) that took place near midday on June 6 and July 27 in 1994, and on July 30, 1996. In addition, at the Northern Study Area (NSA), we analyzed 10 flights over the Old Jack Pine (NSA–OJP) site, 13 flights over the Old Black Spruce (NSA–OBS) site, 8 flights over the Young Jack Pine (NSA–YJP) site, and 8 flights over a site burned in 1979 (NSA–burn). At the Southern Study Area (SSA), we analyzed three flights over the Old Jack Pine (SSA–OJP) site, 9 flights over the Old Black Spruce (SSA–OBS) site, 9 flights over the Old Aspen (SSA–OA) site and one flight over a very recent burn (SSA–recent burn). The NSA–YJP area regenerated following a fire in 1964 (McCaughey et al., 1997), the NSA–OJP stand was 63 years (Gower et al., 1997), the NSA–OBS stand was 120 years (Goulden et al., 1997), the SSA–OJP stand was 65 years (Gower et al., 1997) and the SSA–OBS was 115 years (Gower et al., 1997). Although these flights were made on different days and during different conditions, we analyzed them as a pool of data reflecting growing season conditions. This differs from the analysis of Amiro et al. (1999) where sites were paired using data from the same day. In addition, aircraft flights were conducted

near Inuvik, NWT (67.5 N, 132.9 W) in July of 1999 a few days after a fire where 15- to 25-km long transects were run over burned and unburned areas.

2.2.2. Flux measurements

Details of the Twin Otter operations during BOREAS are fully presented by MacPherson (1996) and MacPherson and Bastion (1997). Briefly, the aircraft is instrumented to measure air motions, geographic position, temperature, dewpoint, solar radiation, net radiation, surface radiometric temperature, some specific radiation wavebands, and concentrations of carbon dioxide, water vapor and ozone. This allows the calculation of fluxes of sensible heat (H), latent heat of evaporation (LE), carbon dioxide, ozone and momentum using the eddy-covariance technique at the aircraft height (MacPherson, 1990). Data were recorded at a rate of 16 samples s^{-1} in 1994 and 32 samples s^{-1} in 1996. At an airspeed of approximately 60 m s^{-1} the minimum resolvable wavelength is about 12 m in 1994 and 6 m in 1996. The flux data were linearly detrended and corrected for time lags between the various sensors (MacPherson, 1996), and included density corrections for the CO_2 fluxes. For the long transect, the runs were divided into 2 km segments, but the mean term in the eddy-covariance calculation was based on a 40 km segment. This reduced the possibility of underestimating the flux but it partially contaminated the flux estimates in cases where different underlying surfaces were experienced along the 40 km segment. The measurements were typically taken at a height of about 30 m above the canopy.

We plotted the polygons for known fires greater than 200 ha in area for the 1980–1996 period along the flight line for the long transect (Amiro et al., 1999). Approximately, 160 segments were classified as vegetated control segments, which had burned prior to 1980. Areas burned in 1995 were separated from control segments, even for the 1994 flights that occurred before the fire. For the NSA and SSA sites, we used the actual site age as the estimate of time since fire, assuming that all sites were generated by fire.

2.3. Remote sensing/NPP modeling

2.3.1. Data sets

We used an annual NPP map for 1994 produced with the Boreal Ecosystem Productivity Simulator (BEPS) model, described by Liu et al. (1997, 1999). This model is modified from FOREST-BGC (Running and Coughlan, 1988) and uses daily meteorological inputs to drive the process-based calculations of NPP. Spatially, the model uses the advanced very-high resolution radiometer (AVHRR) satellite measurements to estimate leaf-area index (LAI) and land-cover type (Chen and Cihlar, 1996; Cihlar et al., 1997) as model inputs. Soil water holding capacity was derived from the database of Shields et al. (1991), and the meteorological inputs were based on a gridded six-hourly weather forecast archive of the US National Center for Atmospheric Research, which were adjusted and validated using data from Canadian climate stations. Total NPP (i.e., above and below ground) is modeled as the gross primary productivity (GPP) minus the autotrophic respiration of all plant components (leaf, stem, root) calculated on a nominal 1 km × 1 km pixel. Partial validation of the model is described by Chen et al. (1999a) and Liu et al. (1997).

2.3.2. NPP with time since fire

We combined a database of fires greater than 200 ha in area (Stocks et al., 2002) with the NPP map of Canada to look at the NPP values following fires of a given year. This was done by overlaying the fire-boundary polygon onto the NPP map within the GIS. The data set has NPP estimated in increments of 5 g C m^{-2} per year and we classed the values into ranges of 50 g C m^{-2} per year, which were then color-coded. Pixels with NPP values of <5 g C m^{-2} per year were invariably classified as water bodies. Therefore, the minimum possible value for a land surface was 27.5 g C m^{-2} per year (i.e., the mean value for the 5–50 g C m^{-2} per year class). The pixels of a given class were counted manually within each fire, and we excluded pixels at the edges of fire boundaries and lakes. Only the most recent burn age was used if fire polygons from different years coincided. This rarely happened for most areas since over-burning was rare. The results were examined by ecozone (Lowe et al., 1996) to group similar conditions.

2.4. Amalgamation of evidence

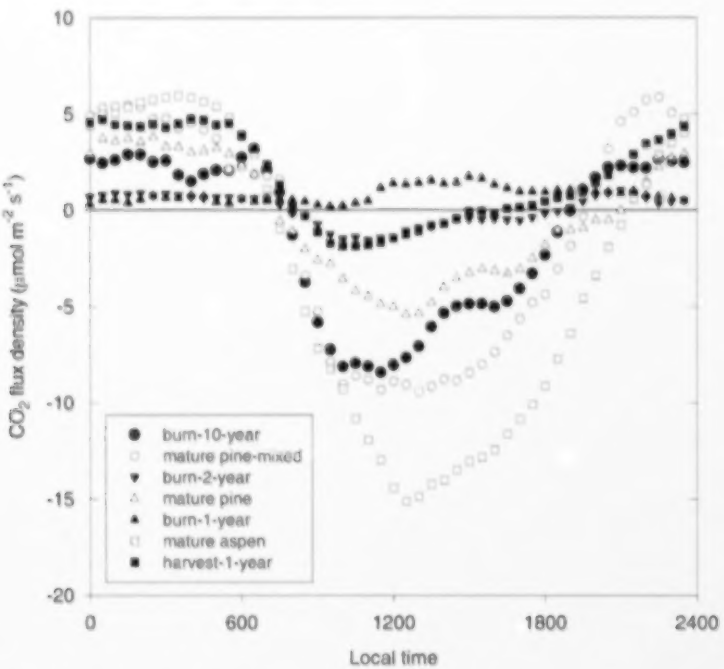
Our goal is to derive relative relationships between the magnitude of the carbon flux and time since fire.

However, the data are from various sources with different spatial and temporal characteristics. For example, the tower data can be derived as daily totals for specific stands, the aircraft data cover the landscape for minutes to hours, and the NPP data are for a full year for large spatial scales. Time since fire is the independent variable, so we normalized the data to a time when data were available from all methods. We did this by calculating the flux value at 10 years since fire using the regression for each data set. For example, for NPP we calculated regressions of NPP vs. time since fire for each ecozone and took the NPP value at 10 years since fire as the normalization value. We then calculated the relative NPP for each ecozone to this 10-year value to estimate a new normalized relationship. Similarly, this was done for the aircraft and tower data separately. The final result is a normalized relationship that allows comparison of a carbon flux value with time since fire for various techniques.

3. Results and discussion

3.1. Tower fluxes

The mean half-hourly values of CO_2 flux density show consistent differences among the seven tower sites (Fig. 3). The mature aspen site shows the strongest daytime uptake and greatest respiration at night with the largest net daily flux of $-3.8\,g\,C\,m^{-2}$ per day (Table 1). These fluxes are similar to those measured at the southern Old Aspen site during summer (Black et al., 1996). The mature pine-mixed and burn-10-year sites are similar in the morning, but the mature-pine-mixed is a stronger sink in the afternoon and a greater source at night. This difference is real since these two sites were operated simultaneously under identical weather conditions. However, the daily integral is identical for the two sites (Table 1). The mature pine site has a lower daytime flux than the other mature sites but also respires less at night.



ig. 3. Tower CO_2 flux data. Data are half-hour means for 1–3-week periods in July or August; the trends were smoothed by filtering each oint using equal weightings for the previous and following hour (i.e., 5-point running filter). Data collected simultaneously are shown s similar symbol types with different shading; paired sites are: burn-1-year and mature pine; burn-10-year and mature pine-mixed; and arvest-1-year and mature aspen.

This stand was at a higher latitude, although mean temperatures were similar to the other sites (Table 1). The data are almost identical to those measured by McCaughey et al. (1997) at the BOREAS NSA–OJP site during June/July 1994, even showing the same afternoon shoulder likely caused by stomatal closure. The burn-2-year site is a small daytime sink with small night respiration (Fig. 3) with about a zero daily net balance (Table 1). Most of the regenerating vegetation is aspen saplings and sparse grass and herbs that create only a small flux. We cannot separate the heterotrophic respiration from the net flux, but it appears from the night-time data that heterotrophic respiration must be small relative to night-time fluxes in more mature stands (Fig. 3). The burn-1-year stand has continuous respiration through the day and night, slightly increasing during the day, probably because of warmer soil temperatures. This stand had almost no vegetation regeneration at this point, so the net flux is dominated by heterotrophic respiration. We have also included data on an aspen site that was harvested one-year previously. This had aspen suckers and other regenerating vegetation that gave daytime fluxes about equal to those at the burn-2-year site (Fig. 3). However, night respiration was much greater, similar to the mature sites, which caused the daily integral flux to be a strong efflux of about 1.6 g C m^{-2} per day (Table 1). We believe that much of this respiration may be autotrophic from the remaining roots of the harvested aspen and poplar trees, but have no independent measurements to confirm this.

3.2. Aircraft fluxes

The fluxes from the aircraft show an efflux during the same year as the fire (SSA-recent burn and Inuvik burn), but downward fluxes in subsequent years (Fig. 4). A general trend towards stronger fluxes is seen although the main effect is within the first 20 years following disturbance. We have plotted the BOREAS long transect flight points and the Inuvik control runs at 50 years although these represent a range of ages greater than 15 years since fire. Also, the SSA-OA site had fluxes of $-15\,\mu mol\,m^{-2}\,s^{-1}$, which we have not

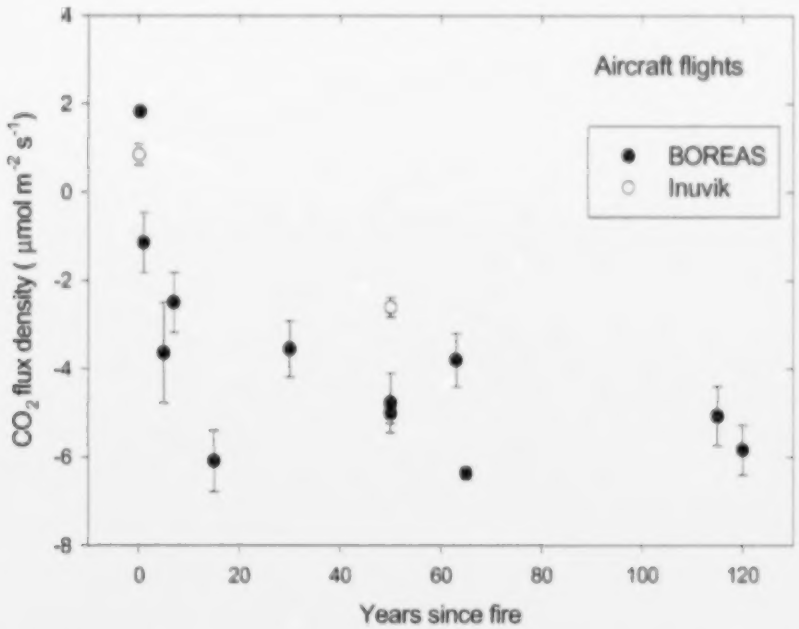


Fig. 4. Aircraft CO_2 flux data (mean ± 1 S.E.) over forests of different ages since fire. Data include flights along the 500 km transect and over stands of known ages during BOREAS in 1994 and 1996, and near Inuvik, NWT in 1999. Points plotted at 50-years-since-fire represent sites of unknown fire age.

plotted in Fig. 4 because of the much greater magnitude of this flux. Previous intercomparisons among aircraft and tower-based fluxes show some variability, with the aircraft giving slightly lower fluxes than the towers, most likely due to flux divergence with height. In general, however, during BOREAS lower sensible and higher latent heat were observed with the aircraft-based system as compared to the tower-based systems (Desjardins et al., 1997). This difference is likely caused by the towers tending to be located in slightly drier areas.

3.3. Remote sensing/NPP modeling

Amiro et al. (2000) present NPP data for different ecoregions across Canada where there have been sufficient numbers of fires to study. Here, we look at the mean values for ecozones (ecozones are larger groupings of several ecoregions) in the western part of the country (Fig. 1). About 70,000 pixels were counted representing 272 fires in the boreal shield ecozone, 25 fires in the boreal plains ecozone, 70 fires in the taiga plains ecozone, and 92 fires in the taiga plains ecozone. Fig. 5 shows increasing NPP with time since fire, with each ecozone having unique characteristics. The taiga shield shows a low NPP with a lower regression slope, as vegetation takes longer to become established in colder climates, especially in areas where soil erosion occurs following fire. The taiga plains ecozone has slightly higher values, likely because of better soil conditions and climate, especially in the Mackenzie River valley. The boreal plains ecozone shows a continuous increase, whereas the boreal plains shows little trend with time since fire. Our data for the boreal plains start at year four, where NPP is already much greater than for the other ecozones. This ecozone has a large deciduous component, mostly aspen, that regenerates following fire and grows quickly. This new vegetation shows readily in the remote sensing data, and if crown closure happens quickly, then we perceive little change in NPP for the following years. Grass and herbaceous species occurring soon

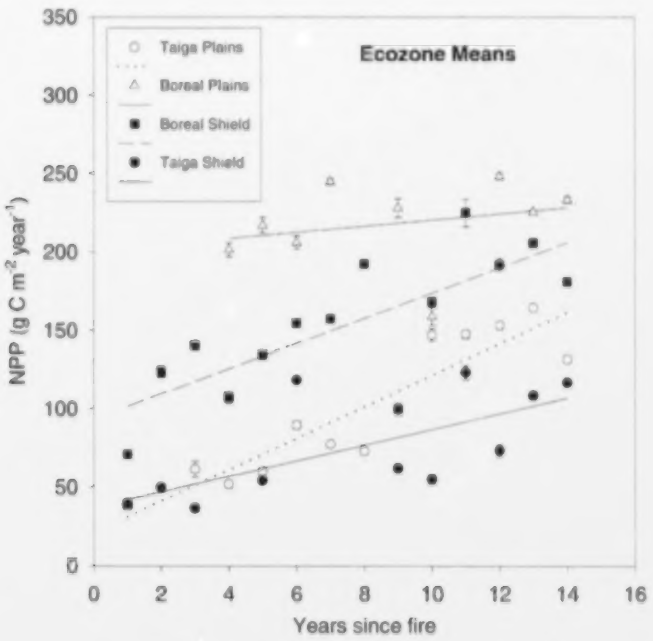


Fig. 5. NPP estimates for four ecozones in western Canada (locations in Fig. 1). Points and error bars are mean ± 1 S.E. Linear regressions on the mean data (for units shown) are NPP = 2 time + 201 for the boreal plains ecozone ($r^2 = 0.07$), NPP = 8 time + 93 for the boreal shield ecozone ($r^2 = 0.59$), NPP = 10 time + 21 for the taiga plains ecozone ($r^2 = 0.85$) and NPP = 5 time + 37 for the taiga shield ecozone ($r^2 = 0.47$).

after the fire may also contribute significantly to NPP in the early post-fire years. Although efforts were made to suppress the background understory effects on retrieval of LAI in forested areas, it was still a factor in the recent burns, which caused the substantial NPP values in the early post-fire years of some ecozones.

3.4. Chronosequence of carbon flux

One of our goals is to estimate NBE annually over a spatial scale sufficient to support modeling efforts. Typically this would be at the ecozone or national level to verify carbon accounting for forests. Fig. 6 illustrates the time and spatial scales covering the measurements in the present study. The tower data give us NEE over typical scales of 0.1 to 2 km, and in our case we have only measured over periods of less than about 20 days at any given site. However, tower flux measurements are being run at many sites for much longer periods to get annual NEE (e.g., Goulden et al., 1996; Black et al., 2000; Valentini et al., 2000). The aircraft data give NEE for short periods over any given spatial sector of a few km, and also give NBE since a large part of the landscape (>100 km) can be covered. The combination of remote sensing data and a model gives NPP and NBP (net biome productivity including all landscape features) over large spatial scales for a full year.

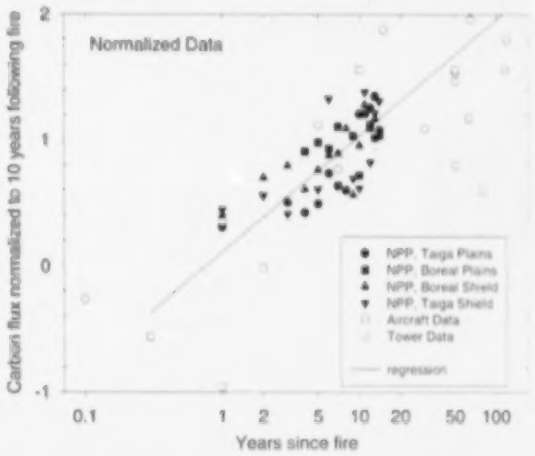


Fig. 7. Carbon flux from different methods normalized to 10 years following fire. The normalization is within each data type (NPP, aircraft, tower). The regression line is relative flux $= 0.11 + 0.92 \log_{10}$ (years since fire), $r^2 = 0.51$.

The various data sources, internally normalized to 10 years after fire, show a reasonably consistent trend, with carbon flux increasing with time since fire (Fig. 7). We have shown all NPP data for the ecozones covering the period from 1 to 14 years following fire. The aircraft and tower data extend to periods beyond this, although there is no clear trend for periods following 20 years after fire. The

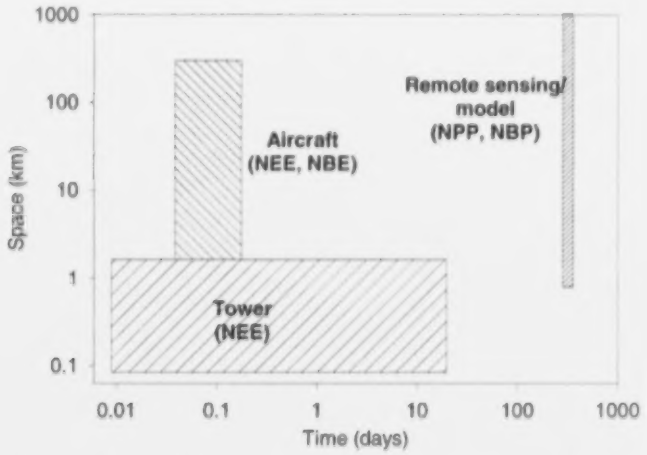


Fig. 6. Space and time scales describing carbon flux measurements in the present study. NEE = net ecosystem exchange; NBE = net biome exchange; NPP = net primary productivity; NBP = net biome productivity.

tower data are the most variable, partly because they cover specific stands with different characteristics (we did not include the mature aspen forest on the graph since it's relative value is about 5). The larger landscape methods (aircraft, remote sensing) blend individual stands. Negative values indicate a net carbon source, which clearly occurs at times less than 1 year after fire. A regression of these data in Fig. 7 is biased towards the larger number of NPP points, but gives a function where the carbon flux relative to 10 years = $0.11 + 0.92 \log_{10}$ (years since fire) with $r^2 = 0.5$. For example, 5 years after fire the carbon flux is typically about 75% of that at 10 years.

The period that it takes for recovery of the carbon flux to the "mature" state is site dependent. For example, the paired tower data for the 10-year-old burned site and the 50-year-old pine-mixed forest are not significantly different in their daily integration for the short measurement period. This appears to be caused by lower photosynthesis and respiration counterbalancing at the burned site (Fig. 3). At this site, most of the fire-killed trees had fallen over, but many were not yet touching the ground so decomposition may play a larger role in future years. The aircraft data suggest that recovery of the daytime carbon flux occurs at about 30 years (Amiro et al., 1999), and the remote sensing/modeling data show NPP recovery at about 20–30 years (Amiro et al., 2000). The consistency in the time trend of NEE and NPP up to about 50 years after fire suggests that the changes in the carbon flux after fire were dominated by vegetation re-growth rather than by the total ecosystem respiration. The reasons are: (i) the remote sensing-based NPP estimates include only the carbon fluxes resulting from re-growth; (ii) the aircraft NEE measurements are affected by both re-growth and soil respiration, and the consistency in the time trends between remote sensing NPP and the aircraft NEE indicates that it is likely that the strength of re-growth dominates the NEE increase with time since fire and (iii) the tower NEE data also indicate the similar trend, although the data are more scattered.

Fig. 7 is internally consistent among the data sets. However, there are differences in time of year that contribute to the scatter, especially since the greatest CO_2 fluxes are typically in the early growing season, slowly decreasing through the summer (Black et al., 2000). The sampling over large spatial scales by all methods includes more variability than that is normally expected because of different weather conditions at any given site. However, heterotrophic respiration is not measured by the remote sensing technique, nor is it fully incorporated into the aircraft sampling (no night-time measurements) or into the tower sampling (no winter data). A full accounting of carbon exchange for each of these methods needs to include all respiration terms. Hence, a separate evaluation of the potential trends in respiration response to fires follows.

3.5. Respiration issues

Change in ecosystem respiration with time since fire is potentially an important part of the net carbon flux. Most of the respiration is from the soil, but this includes heterotrophic respiration (largely by soil microbiota) and autotrophic respiration by plant roots. In many cases, only total soil respiration has been measured. A review by Johnson (1992) concluded that low-intensity burns had very little effect on soil respiration. Weber (1985) found a varied response in jack pine stands, depending on the age and nature of the burn, but a stand-replacing fire that occurred during early stages of stand development significantly lowered the respiration rate 6 years after the burn. Also in a boreal aspen stand, burned plots had decreased respiration compared to control plots for the first two seasons with recovery in the third (Weber, 1990). During the BOREAS experiment, Burke et al. (1997) measured respiration in black spruce sites of different ages and found no effect immediately after fire, decreased respiration 2 years later, with recovery to pre-burn levels after 7 years. At Russian dwarf-shrub tundra sites, 2-year-old and 8-year-old burned sites were carbon sinks compared to a control site, which was a carbon source (Zamolodchikov et al., 1998). For peat soils, Hogg et al. (1992) suggest that removal of the top peat layers by fire will decrease total respiration because the deeper layers respire more slowly. In contrast to these forest studies, Knapp et al. (1998) found increased respiration in burned grassland sites of the Konza Prairie for the first 2 years.

This general decrease in total soil respiration can be partially explained by a decrease in autotrophic respiration when vegetation is killed by the fire. Sawamoto et al. (2000) found decrease in total soil respiration, but increase in heterotrophic respiration following

fire in Siberia, because more than half of the total soil respiration was by tree roots (i.e., autotrophic). Similarly, Richter et al. (2000) found that total soil respiration following fires in Alaska was about half of the control site respiration, but reasoned that this was caused by decreased autotrophic respiration, and that heterotrophic respiration likely increased. However, many other studies show a decrease in post-fire heterotrophic respiration. A study using litter bags in a jack pine forest showed that decomposition rates of understory samples were slower in newer stands than in older ones (Weber, 1987). In a boreal jack pine site, Ahlgren and Ahlgren (1965) found a decrease in respiration from collected soil samples immediately following a burn, but recovery soon happened such that there was little difference compared to a control. Bissett and Parkinson (1980) measured respiration in transplanted media 6 years following a wildfire in a spruce–fir subalpine forest: in laboratory tests, the burned site had lower respiration, but in the field, the burned site respired more. They believed that this difference was caused by warmer temperatures at the burned field site compared to the control site. Following a prescribed fire in a ponderosa pine stand, respiration was not immediately affected (on an organic matter basis), but the decomposition rate was lower at the burned site for a period of 6–10 months following the burn (White, 1986). In Finland, in areas of prescribed burns of Scot's pine, decomposition of pine needles in litter bags increased even though total respiration decreased (Fritze et al., 1994; Pietikainen and Fritze, 1996). Pietikainen and Fritze (1993) followed soil respiration in prescribed burned areas that were clear-cut slash and non-cut, and found that over 3 years, soil respiration was lower in the burned sites compared to the controls. In a study of several sites ranging to 45 years, Fritze et al. (1993) found that soil respiration decreased in the first 2 years, increased to a plateau around 12 years, then decreased at about 40 years. When the soils were adjusted to a higher water content, then the respiration difference was lessened between the treatments (Fritze et al., 1994) or the total respiration in burn plots actually increased over the controls (Pietikainen and Fritze, 1996). The explanation for these differences was that moisture was limiting in the burn plots.

A potential increase in soil respiration is often attributed to increased soil temperatures following fire. Short-term temperature increases often increase respiration with Q_{10} values often assumed to be about 2 (Katterer et al., 1998), although many forest studies suggest values of 4 or 5 (Black et al., 1996; Davidson et al., 1998; Russell and Voroney, 1998). However, Giardina and Ryan (2000) summarized data from a wide geographic area showing that decomposition rates are remarkably constant, despite wide variations in mean annual temperature. The question is whether local soil respiration quickly acclimatizes to the new regime following fire, such that temperature alone does not have a large effect. One of the explanations for the apparent decrease in heterotrophic respiration is the loss of microbes. Microbial biomass has been shown to decrease for 3 years following fire in a Scot's pine forest (Fritze et al., 1994), and bacteria first decreased and then increased following a fire in a jack pine community (Ahlgren and Ahlgren, 1965). In a loblolly pine forest, the total fungal mass decreased because of the loss of substrate although the density of fungi did not change (Jorgensen and Hodges, 1970). Also, experimentally applied heat has been shown to reduce fungi in the soil (Pattinson et al., 1999).

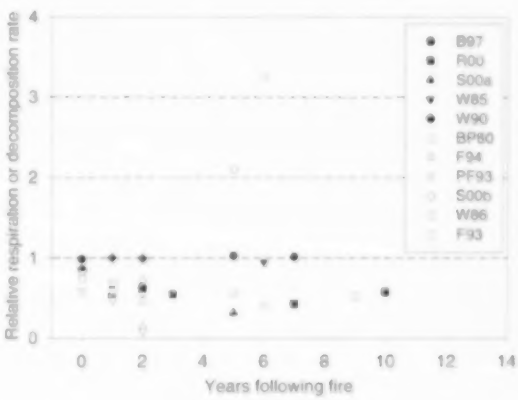


Fig. 8. Literature measurements of respiration or decomposition rates with time since fire, expressed relative to control plots or the condition before fire (i.e., values <1 indicate respiration decrease). Total soil respiration measurements are solid symbols where B97—Burke et al. (1997), R00—Richter et al. (2000), S00a—Sawamoto et al. (2000) including roots, W85—Weber (1985), W90—Weber (1990). Decomposition or respiration without roots (heterotrophic) are open symbols where BP80—Bissett and Parkinson (1980), F94—Fritze et al. (1994), PF93—Pietikainen and Fritze (1993), S00b—Sawamoto et al. (2000) without roots, W86—White (1986), F93—Fritze et al. (1993).

The microbial composition (bacteria and fungi) also changes following fire (Bissett and Parkinson, 1980), and this is expected to alter the respiration rate.

We have summarized these studies in Fig. 8 by comparing either pre-fire to post-fire situations, or post-fire to control plots for the various studies. Soil respiration generally decreases following forest fire. This is caused by decreased autotrophic respiration, but decomposition also decreases, probably caused by a decreased microbial population and less carbon substrate. This decrease in respiration contrasts with results of the model of Auclair and Carter (1993), which estimated that 60% of the total release of CO_2 from fire was caused by enhanced soil respiration. We still have some uncertainty here, and regional estimates of respiration might be feasible in the future using a combination of aircraft-based CO_2 flux measurements and remote-sensing data (Desjardins et al., 1995).

4. Conclusions

The data from towers, aircraft, and remote sensing/modeling measure different components of the carbon flux over a range of spatial and temporal scales. However, in all cases, the measurements show that fire reduces the net downward flux, which then increases over time for a period of 10–30 years following fire. Upward net fluxes are only apparent immediately (i.e., within 1 year) after fire and harvesting, and photosynthesis is greater than respiration during the day for the growing season, even during early succession. Although the daily summer-time integral of NEE from towers, the afternoon values of NEE from aircraft, and the annual NPP estimates from remote sensing/modeling show similar trends with time since fire, we do not yet have annual estimates of NEE. It is likely that heterotrophic respiration during the non-growing season would reduce the annual NEE, perhaps changing the balance to a net carbon source for several years following disturbance. However, this is still unknown, and the bulk of respiration studies in the literature suggest that total respiration (autotrophic and heterotrophic) usually decreases following fire, although the effect on heterotrophic respiration alone is not always clear. This will impact some carbon balance models that assume increased heterotrophic respiration following fire (e.g., Auclair and Carter, 1993; Kurz and Apps, 1999). When annual measurements of NEE following disturbance become available, we hope to be able to provide data to help verify and improve carbon balance models that include the effect of disturbance in the boreal forest.

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